

Coupling Between the Coastal Ocean and Yaquina Bay, Oregon: Importance of Oceanic Inputs Relative to Other Nitrogen Sources

Cheryl A. Brown · Robert J. Ozretich

Received: 19 October 2007 / Revised: 2 December 2008 / Accepted: 3 December 2008 / Published online: 16 January 2009
© U.S. Government 2009

Abstract Understanding of the role of oceanic input in nutrient loadings is important for understanding nutrient and phytoplankton dynamics in estuaries adjacent to coastal upwelling regions as well as determining the natural background conditions. We examined the nitrogen sources to Yaquina Estuary (Oregon, USA) as well as the relationships between physical forcing and gross oceanic input of nutrients and phytoplankton. The ocean is the dominant source of dissolved inorganic nitrogen (DIN) and phosphate to the lower portion of Yaquina Bay during the dry season (May through October). During this time interval, high levels of dissolved inorganic nitrogen (primarily in the form of nitrate) and phosphate entering the estuary lag upwelling favorable winds by 2 days. The nitrate and phosphate levels entering the bay associated with coastal upwelling are correlated with the wind stress integrated over times scales of 4–6 days. In addition, there is a significant import of chlorophyll *a* to the bay from the coastal ocean region, particularly during July and August. Variations in flood-tide chlorophyll *a* lag upwelling favorable winds by 6 days, suggesting that it takes this amount of time for phytoplankton to utilize the recently upwelled nitrogen and be transported across the shelf into the estuary. Variations in water properties determined by ocean conditions propagate approximately 11–13 km into the estuary. Comparison of nitrogen sources to Yaquina Bay shows that the ocean is the

dominant source during the dry season (May to October) and the river is the dominant source during the wet season with watershed nitrogen inputs primarily associated with nitrogen fixation on forest lands.

Keywords Ocean input · Upwelling · Nutrient sources · Yaquina estuary

Introduction

In most estuaries, the major sources of nitrogen are atmospheric deposition, agricultural nitrogen fixation, fertilizer runoff, and in heavily populated areas point source inputs associated with wastewater treatment facilities (Boyer et al. 2002; Howarth et al. 2002; Driscoll et al. 2003). For many estuaries in the Pacific northwest (PNW) of the United States, there are relatively low population densities in the watersheds and low atmospheric deposition rates. Land use in the watersheds is predominantly forested, resulting in low nitrogen (N) inputs associated with fertilizer and agriculture N fixation. In addition, upwelling provides nutrients to estuaries adjacent to coastal upwelling regions, such as the PNW (e.g., Hickey and Banas 2003). The differences in land use combined with coastal upwelling may result in differences in the dominant N sources to PNW estuaries compared to other regions.

In a recent review, Tappin (2002) found that the N input to temperate and tropical estuaries associated with the ocean is poorly quantified. Previous studies have demonstrated that the oceanic inputs of nutrients and phytoplankton are important for estuaries adjacent to coastal upwelling regions, such as the west coast of the United States (e.g., de Angelis and Gordon 1985; Roegner and Shanks 2001; Roegner et al. 2002; Colbert and McManus 2003). It is important to quantify the contribution of oceanic input to nutrient loading in order to determine reference conditions for estuaries

C. A. Brown (✉)
Pacific Coastal Ecology Branch, Western Ecology Division,
U.S. Environmental Protection Agency,
2111 S.E. Marine Science Center Drive,
Newport, OR 97365, USA
e-mail: brown.cheryl@epa.gov

R. J. Ozretich
Western Ecology Division,
U.S. Environmental Protection Agency,
200 SW 35th,
Corvallis, OR 97333, USA

adjacent to upwelling regions and to distinguish natural variability from anthropogenic inputs. In addition, we do not know how susceptible estuaries subjected to large oceanic inputs of nutrients (dissolved inorganic nitrogen and phosphorous) are to future changes in anthropogenic inputs of nutrients. Some studies have suggested that future climate change may lead to changes in the seasonality or intensity of wind-driven upwelling (Snyder et al. 2003), which could modify the nutrient loading to these systems; therefore, it is important to quantify the oceanic input of nutrients to establish a baseline.

Previous studies of the importance of oceanic variability in estuarine water properties consisted of short-term observations of nutrients or chlorophyll *a* (de Angelis and Gordon 1985; Roegner and Shanks 2001) or examined water temperature or salinity fluctuations (Hickey et al. 2002; Hickey and Banas 2003). De Angelis and Gordon (1985) demonstrated that there was import of oceanic dissolved inorganic nitrogen to Alsea Bay, Oregon; however, they only sampled on six dates during the summer of 1979 and only two of those dates had significant oceanic import of nitrate (NO_3^-). Roegner and Shanks (2001) demonstrated chlorophyll *a* was imported into an Oregon estuary from the coastal ocean; however, they had insufficient temporal resolution in their data to examine the coupling between wind stress and chlorophyll *a*. Hickey et al. (2002) demonstrated that water property fluctuations near the mouth of Willapa Bay, WA, USA, are related to alongshore wind stress and propagate up the estuary, but their study focused on temperature, salinity, and current velocities.

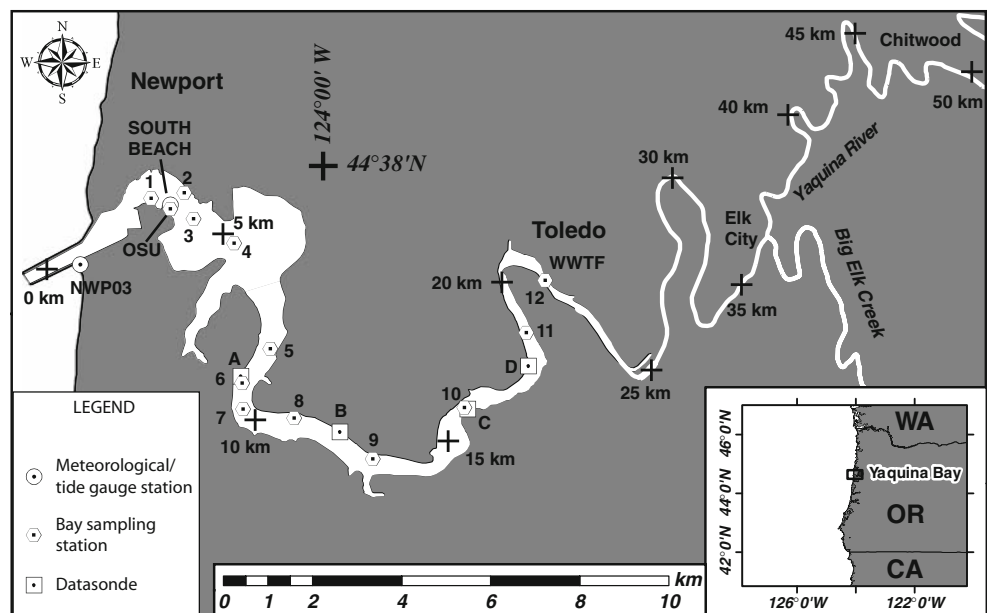
High temporal resolution data are required to demonstrate the coupling between wind stress and water column

properties (nutrients and chlorophyll *a*) and to quantify the gross oceanic loading. In this paper, we quantify the gross oceanic input of nutrients and chlorophyll *a* using data collected daily during the upwelling season (May–September) for two consecutive years. In addition, we examine the coupling between wind forcing and nutrient and phytoplankton levels entering the estuary and the propagation of these signals into the estuary. We also compare the major N inputs to the estuary (including gross oceanic, riverine, wastewater treatment facility effluent, benthic flux, and atmospheric inputs) to assess the importance of oceanic inputs relative to other sources.

Study Location

Yaquina Bay is a small drowned river estuary located along the central Oregon coast of the United States (Fig. 1) with a surface area of 13 km² and a watershed surface area of 658 km² (Quinn et al. 1991). The Yaquina watershed is primarily forested (94.7%) with urban and agricultural activities occurring on 2.9% and 1.6% of the watershed, respectively (<http://cads.nos.noaa.gov/>). This bay experiences mixed semidiurnal tides with mean tidal range of 1.9 m and a tidal prism volume of $2.4 \times 10^7 \text{ m}^3$ (Shirzad et al. 1989). Due to the small volume of the estuary ($2.5 \times 10^7 \text{ m}^3$ at mean lower low water) and the strong tidal forcing, there is close coupling between the estuary and the coastal ocean. About 70% of the volume of the estuary is exchanged with the coastal ocean during each tidal cycle (Karentz and McIntire 1977). Yaquina Bay receives freshwater inflow primarily from two tributaries, the Yaquina River and Elk Creek, which have similarly sized drainage areas and

Fig. 1 Map of study area showing the location of ocean input sampling location (OSU), bay sampling stations (Stations 1–12), water quality monitoring datasondes (Stations OSU, A, B, C, and D), and meteorological and tide gauge stations (South Beach and NWP03)



contribute approximately equally to the freshwater inflow (State Water Resources Board 1965). During November through April, the Oregon coast receives high precipitation and the estuary is river dominated. Approximately 77% of the total annual precipitation of 68 in. occurs during November through April (<http://www.wrcc.dri.edu>, calculated using long-term statistics for Newport, Oregon). From May through October, there is a decline in the riverine freshwater inflow and the estuary switches from riverine to marine dominance. The estuary is classified as well-mixed under low flow conditions and as partially mixed during winter high riverine inflow conditions (Burt and McAlister 1959). The flushing time of the estuary during the summer varies from 1 day near the mouth to 9 days in the upstream portions (Choi 1975). During the summer, winds from the north drive coastal upwelling, which brings cold, nutrient rich waters to the surface that enter the estuary during flood tides. In addition to the riverine and oceanic nutrient inputs to the system, the City of Toledo, Oregon (population of approximately 3,400; source: 2004 Census, <http://www.census.gov>) discharges wastewater treatment facility effluent into the Yaquina Bay 22 km upstream of the estuary mouth. The City of Newport, Oregon (population of approximately 9,600; source: 2004 Census, <http://www.census.gov>) is located adjacent to Yaquina Bay, however, wastewater effluent from this community is discharged 2 km offshore.

Materials and Methods

Oceanic Input

During May through October of 2002 and 2003, daily water samples were collected during flood tides approximately 0.5 m below the surface at the Oregon State University Dock (labeled OSU in Fig. 1), which is located inside the bay 4 km from the seaward end of the jetties. The samples were immediately filtered and frozen for storage until analysis. Dissolved inorganic nutrients ($\text{NO}_3^- + \text{NO}_2^-$, NH_4^+ , PO_4^{3-} , and H_4SiO_4) were analyzed by MSI Analytical Laboratory, University of California-Santa Barbara, CA using Lachat flow injection instrumentation (Zellweger Analytics, Milwaukee WI, USA). One-liter surface water samples were collected daily and analyzed for chlorophyll *a*. These samples were filtered within 15 min using 47-mm diameter GF/F filters. Chlorophyll *a* was extracted by sonicating the filters and soaking them overnight in 10 ml of 90% acetone. The next morning the samples were centrifuged and analyzed for chlorophyll *a* content using a fluorometer (10 AU Fluorometer, Turner Designs, Inc, Sunnyvale, CA, USA). Beginning on July 23, 2002, an in situ fluorometer (SCUFA, Turner Designs, Inc., Sunnyvale, CA, USA) was deployed at the OSU dock

providing in situ fluorescence. Commencing on August 28th 2002, an automated water sampler (ISCO®, Model 3700FR, Lincoln, NE, USA) was used to collect water samples for each flood tide and programmed using the predicted time of each high tide. The sampler held the samples in a dark refrigerated compartment and the samples were collected daily, filtered, and frozen for nutrient analyses.

Physical Data

Hourly wind speed and direction data were available from nearshore and offshore stations adjacent to Yaquina Bay (NWP03 and 46050, respectively) operated by the National Data Buoy Center (<http://www.ndbc.noaa.gov>). Station NWP03 is located at the entrance to Yaquina Bay (Latitude 44.61° North, Longitude 124.07° West; Fig. 1) and Station 46050 is located at the 130-m depth contour 36 km offshore of Yaquina Bay (Latitude 44.62° North, Longitude 124.53° West). Gaps in the wind data of less than 6 h were filled using linear interpolation. During 2002, there was a gap of approximately 2 days in wind data from Station 46050 that was filled using the relationship between north-south wind stress at 46050 and NWP03. Alongshore wind stress (τ_y) was computed using the method of Large and Pond (1981) with a positive wind stress indicating upwelling favorable wind stress from the north. For the correlation analysis between wind stress and water column parameters, we used average daily and integrated alongshore wind stress. The integrated alongshore wind stress (W_k) was calculated as a weighted running mean of the wind stress which weights the past alongshore wind stress with a decaying exponential function (Austin and Barth 2002). The integrated alongshore wind stress at time (T) is defined as

$$W_k(T) = \int_0^T \frac{\tau_y}{\rho} e^{(t-T)/k} dt \quad (1)$$

where τ_y is the alongshore windstress at time t , ρ is seawater density, and k is an exponential decay coefficient. Equation 1 is integrated with $t=0$ defined as January 1 of each year. The weighting function used in the calculation of the running mean has an e-folding decay scale of k . Correlation analysis was performed for values of k ranging from 0 to 50 days.

Water temperature, tide height, wind speed, and direction were obtained from South Beach Station (Station: 9435380, latitude 44.625° North, longitude 124.043° West, location presented in Fig. 1) operated by the Center for Operational Oceanographic Products of the National Ocean Service (<http://co-ops.nos.noaa.gov>). Flood tide water temperatures were extracted from the hourly data using times of predicted high tides. Data from South Beach Station were used because it had minimal data gaps. Water temperature

data from the South Beach Station were compared to two YSI, Inc. (Yellow Springs, OH, USA) Multiparameter Monitoring Systems located at Station OSU (Fig. 1), which were deployed near the surface (about 1 m below water surface) and a second at an average depth of 2 m below water surface, as well as data from the bay sampling (see Station 1 in next section). During 1999–2002, there was close agreement between the time series of water temperature at South Beach and other data sources; however, during 2003, the South Beach water temperature was approximately 1°C colder than the OSU Station, so we adjusted (added 1°C) the South Beach time series during this year. Water temperature and salinity were available at four other locations from YSI datasondes (Specht, unpublished data) that were deployed at Riverbend (Station A), Oregon Oyster (Station B), Cragie Point (Station C), and Criteser's Landing (Station D) (Fig. 1). The datasondes at Riverbend and Cragie Point (Stations A and C) were deployed at an average depth of about 4 m and 2 m below the water surface, respectively, while the datasondes at Oregon Oyster and Criteser's Landing (Stations B and D) were deployed at a depth of about 1 m below the water surface. All variables were logged every 15 min.

Bay Sampling

Twelve locations in the estuary were sampled at approximately weekly intervals for dissolved inorganic nutrients at mid-depth and 0.5 m above the bottom (locations shown as Stations 1–12 in Fig. 1). Water samples were collected from depth using a hand-operated pump, filtered (45 µm filter) and frozen until analysis. The samples were analyzed for $\text{NO}_3^- + \text{NO}_2^-$, NH_4^+ , H_4SiO_4 , and PO_4^{3-} . At each station profiles of conductivity, temperature and depth (CTD; SBE 19 SEACAT Profiler, Sea-Bird Electronics, Inc, Bellevue, WA, USA) and in situ fluorescence (WETStar Chlorophyll Fluorometer, WET Labs, Philomath, OR, USA) were measured. The profile measurements were taken at 0.5-s intervals from the water surface to 0.5 m above the bottom and during post-processing the data were binned into 0.25-m intervals. The fluorometer was calibrated by collecting water samples quarterly, filtering them, and analyzing them for chlorophyll *a* using the same technique used for the oceanic input samples, and developing a relationship between in situ fluorescence and extracted chlorophyll *a* values ($\text{Chlorophyll } a \text{ (}\mu\text{g l}^{-1}\text{)} = 0.52 \times \text{in situ fluorescence} - 1.75$, $r^2 = 0.95$, $n = 36$). In this equation, in situ fluorescence refers to the factory calibration estimate of chlorophyll *a*. These cruises were conducted during flood tides, tracking the propagation of the tide up the estuary, and were completed in about 3 h. Time series of salinity were examined at Stations A, B, C, and D to confirm that the cruises were tracking the propagation of the tide.

Data Analysis

To examine the relationship between shelf upwelling dynamics and nutrient and phytoplankton entering the bay, we performed a cross-correlation analysis between average daily north–south wind stress (at Stations NWP03 and 46050), flood tide water temperature, dissolved inorganic nutrients, chlorophyll *a*, and in situ fluorescence (at Station OSU) entering the bay. In situ fluorescence data were low-pass filtered using a 3-h Lanczos filter and flood tide values were extracted using times of predicted high tides. Cross-correlation coefficients were calculated using the non-parametric Spearman rank-order correlation coefficient using SigmaStat (version 3.10 Systat Software Inc., Point Richmond, CA, USA). An adjusted sample size (N^*) based on the modified Chelton method (Pyper and Peterman 1998) was used in the correlation analysis to adjust for the effect of autocorrelation in the time-series on significance levels. Gaps in the time series were filled with linear interpolation (time step of 1 day), since the modified Chelton method requires no gaps in the time series. These interpolated time series were used only to determine the adjusted sample size for use in determining the significance levels, but were not used in the calculation of the correlation coefficients.

To examine how far into the bay the nutrient and chlorophyll *a* temporal variability is determined by ocean conditions, we examined the correlation between the $\text{NO}_3^- + \text{NO}_2^-$, PO_4^{3-} , and chlorophyll *a* concentrations found at Station 1 near the mouth of the estuary (Fig. 1) and those stations further in (Stations 2–12) during the period of May through August. During this period, the mean absolute difference between mid-depth and bottom samples for $\text{NO}_3^- + \text{NO}_2^-$, NH_4^+ , and PO_4^{3-} was 0.5, 0.3, and 0.1 µM ($n = 202$), respectively; therefore, we averaged mid-depth and bottom samples for this analysis.

Other Data Used for Comparison of Nutrient Input

The riverine contribution to N inputs to Yaquina Bay was calculated using observations of streamflow and stream nutrient concentrations. The Yaquina River has been gauged by the US Geological Survey and the State of Oregon Water Resources Department at a station near Chitwood, Oregon (USGS Station 1430600), which is 51 km upstream from the mouth of Yaquina Bay. We compared the N sources to Yaquina Bay during the wet and dry seasons. The wet season (November–April) was defined as months when the monthly average discharge of the Yaquina River at Chitwood (computed using data from 1972 to 2002) exceeded the 30-year average discharge of $7.2 \text{ m}^3 \text{ s}^{-1}$, while the dry season (May–October) was defined as months when the monthly average discharge was less than the long-term average.

The program LOADEST (Runkel et al. 2004) was used to estimate the riverine load using a linear regression model. This program takes into account retransformation bias, data censoring and non-normality which complicate load estimations. Stream nutrient data at Chitwood were available from the Oregon Department of Environmental Quality (<http://www.deq.state.or.us>). Using all available dissolved inorganic nitrogen (DIN) data from 1979 to 2005, we generated a relationship between load and streamflow using the USGS program LOADEST (Runkel et al. 2004) with the following terms:

$$\ln(L_{\text{Chitwood}}) = a_0 + a_1 \ln Q + a_2 \ln Q^2 \quad (2)$$

where L_{Chitwood} is the load at Chitwood in kg day^{-1} and $\ln(Q)$ equals $\ln(Q_c)$ minus center of $\ln(Q_c)$, where Q_c is the discharge at Chitwood in $\text{ft}^3 \text{s}^{-1}$ ($r^2=0.98$, $n=87$). The $\ln(Q)$ terms are centered to eliminate collinearity between the linear and quadratic terms of Eq. 2, for more details see Runkel et al. (2004). The regression coefficients (a_0 , a_1 , and a_2) are determined by adjusted maximum likelihood estimation. Loads were estimated for the interval of May 1, 1980–April 30, 2005 using daily discharge data and the regression model. There are two main tributaries to the Yaquina Estuary, the Yaquina River and Elk Creek and the confluence of these two tributaries is at Elk City. The basin ratio method was used to account for the ungauged portions of the watershed with the load at Elk City ($L_{\text{Elk City}}$) estimated as

$$L_{\text{Elk City}} = L_{\text{Chitwood}} \frac{A_{\text{Elk City}}}{A_{\text{Chitwood}}} = 2.52 \cdot L_{\text{Chitwood}} \quad (3)$$

where $A_{\text{Elk City}}/A_{\text{Chitwood}}$ is the ratio of the watershed area at Elk City to that at Chitwood. This estimation of the ungauged portions of the watershed is only valid if the freshwater inflow per unit area of the gauged portion of the watershed is the same as for the ungauged portion and the DIN levels in the two tributaries are similar. Previous studies have shown that the DIN levels in Elk Creek and the Yaquina River are approximately equal (Sigleo and Frick 2007). In addition, limited streamflow measurements at Elk City indicate that flow at Elk City is about 2.2 times that at Chitwood ($n=39$).

The wastewater treatment facility input of DIN to the estuary was computed by multiplying the daily volume discharged by the effluent concentration. Data on daily volume discharge and concentration were provided by the City of Toledo, Oregon Wastewater Treatment Facility for 2002–2004. From the analysis of split samples by UCSB the treatment facility NO_3^- concentrations were found to be biased high and were adjusted (multiplied by 0.78) prior to computing loading. From December 2000 through June 2002 approximately 75% of the discharged nitrogen was NO_3^- ; 15% and 10% as NH_4^+ and organic N, respectively.

Atmospheric N deposition is monitored by the National Atmospheric Deposition Program (NADP) at a station 40 km away from Yaquina Estuary (Alsea Guard Ranger Station, OR02). The annual atmospheric input at this site averaged over the interval of 1980 to 2002 was used to estimate the atmospheric N input to the estuary (NADP 2003). The atmospheric N input on the watershed was calculated as the product of deposition rate ($\text{kg N ha}^{-1} \text{year}^{-1}$) and watershed area, and the direct input to the estuary was calculated as the product of deposition rate and the estuary area.

We quantified the gross oceanic input of DIN to the estuary. We did not have adequate data to quantify the net oceanic input of DIN. The gross oceanic input of DIN was calculated using the time-series of flood tide DIN concentration multiplied by the volume of water entering the inlet during each tidal cycle. The volume of water entering the inlet was calculated using a two-dimensional, laterally averaged hydrodynamic, and water quality model (Cole and Wells 2000). In the model simulations, Yaquina Estuary was represented by 325 longitudinal segments spaced approximately 100-m apart with each longitudinal segment having 1-m vertical layers. The model domain extended from the tidal fresh portion at Elk City, Oregon to the mouth of the estuary. Riverine freshwater inflow was included using a relationship between Chitwood discharge and Elk Creek. Meteorological forcing was included in the model using hourly wind speed and direction data from the South Beach Station and air temperature and dewpoint data from the Hatfield Marine Science Center, Oregon State University, weather station. Water surface elevation from the South Beach Station provided the tidal forcing in the model. Oceanic variations in water temperature were included as a boundary condition using data from the South Beach Station. Water surface elevation data were available at two locations for model calibration (Stations A and C in Fig. 1). In addition, water temperature and salinity data were available at four locations (Stations A, B, C, and D, Fig. 1) for model calibration. Simulations were performed for 2002 and a relationship between modeled volume of water entering the bay each flood tide and difference between low and high tide elevation was developed.

$$Q_{\text{tide}} = (\text{PWL}_{\text{high}} - \text{PWL}_{\text{low}}) \cdot 7.3 \times 10^6 + 0.4 \times 10^6 \quad (4)$$

$$(n = 704, r^2 = 0.99)$$

where Q_{tide} is the volume of water entering the inlet during the flood tide (m^3), PWL_{high} is the tidal elevation at high tide (m, relative to mean lower low water), PWL_{low} is the tidal elevation during the previous low tide (m, mean lower low water) and the standard error of the slope and intercept are 3.0 and 5.7×10^4 , respectively. The location

where Q_{tide} was calculated is approximately 1.5 km from the model's seaward boundary. Tide tables were used to obtain values of PWL_{high} and PWL_{low} for each flood tide during the period of May through the end of September of 1997–2003. We used observations of DIN from our flood tide sampling (2002 and 2003) to estimate the dry season gross oceanic N input. To estimate the DIN in oceanic water entering the inlet during years when we did not have data available, we developed a relationship between dry season water temperature and $\text{NO}_3^- + \text{NO}_2^-$ using data from 1997–2004 (Wetz et al. 2005) from the inner continental shelf off of Newport, OR, USA (Fig. 2). Flood tide $\text{NO}_3^- + \text{NO}_2^-$ (μM) was modeled as

$$\text{NO}_3 + \text{NO}_2 = \frac{1.37 - 44.7}{1 + e^{(T-8.29)/(-0.996)}} + 44.7 \quad (5)$$

($n = 570, r^2 = 0.85$)

where T is water temperature ($^{\circ}\text{C}$). The average value of NH_4^+ in flood tide water during 2002 and 2003 ($3.6 \mu\text{M}$, $n=463$) was added to the modeled $\text{NO}_3^- + \text{NO}_2^-$ to account for this species. The gross oceanic DIN input for the dry seasons of 1997–2003 was estimated using Eqs. 4 and 5 for each flood tide and the dry season mean was calculated for each year. Flood tide water temperature data from South Beach Station (Fig. 1) was used to estimate gross oceanic DIN input due to the minimal number of gaps in this time series. Flood tide water temperature at South Beach are correlated with 40-h low-pass filtered inner shelf water temperature (unpublished analysis, C. Brown). The root mean square error (RMSE) in using Eq. 5 to

estimate the flood tide concentration of $\text{NO}_3^- + \text{NO}_2^-$ was calculated as

$$\text{RMSE} = \sqrt{\frac{\sum_{n=1}^M (N_{\text{observed}} - N_{\text{modeled}})^2}{M}} \quad (6)$$

where M is the number of observations (number of flood tide samples), N_{observed} and N_{modeled} is the observed and modeled $\text{NO}_3^- + \text{NO}_2^-$.

The wet season gross oceanic DIN input was estimated using the average wet season (from November 1997–April 2003) surface DIN at an innershelf station off Newport, OR (Wetz et al. 2005) and modeled amount of water entering each flood tide during the wet season of 2002. The wet season average DIN on the inner shelf was $3.3 \mu\text{M}$ ($n=17$) and the average salinity was 32.1 psu. Wet season mixing diagrams (from 1998–2003) from the Yaquina Estuary were used to confirm the wet season average oceanic DIN. Since mixing diagrams generated from estuary data were often influenced by freshwater inflow the mixing diagrams were extrapolated to salinity of 32.1 psu to estimate the oceanic DIN. The average wet season DIN from the innershelf was consistent with extrapolation of wet season mixing diagrams estimate (average= $4.0 \mu\text{M}$, $n=32$). To estimate the importance of benthic flux on DIN concentrations within the bay, we used published values from Yaquina Bay (De Witt et al. 2004).

Results and Discussion

Flood Tide Input from Ocean

1. Flood tide sampling

a. Dissolved inorganic nutrients

During upwelling conditions of 2002 and 2003, maximum NO_3^- and PO_4^{3-} levels in flood tide water in the lower estuary (Station OSU) were 31.5 and $2.9 \mu\text{M}$, respectively. These maximal nutrient concentrations entering Yaquina Bay during upwelling periods are similar to those found in other upwelling regions (Dugdale 1985) as well as those found on the Oregon shelf (Corwith and Wheeler 2002). During 2002, the flood tide $\text{NO}_3^- + \text{NO}_2^-$ at Station OSU was correlated with $\text{NO}_3^- + \text{NO}_2^-$ measured on the inner-shelf 5 miles off of Newport ($r=0.74$, $n=15$, $p<0.05$, Pearson Product; unpublished data of W. Peterson). During the upwelling seasons of 2002 and 2003, the $\text{NO}_3^- + \text{NO}_2^-$ concentrations in flood tide water entering the estuary ranged from 0.0 to $31.5 \mu\text{M}$ ($11.3 \pm 8.8 \mu\text{M}$, (mean \pm SD), $n=463$), while NH_4^+ concentrations ranged from 0.4 to $9 \mu\text{M}$ ($3.6 \pm 1.7 \mu\text{M}$, $n=463$). Nitrite was a minor component of DIN,

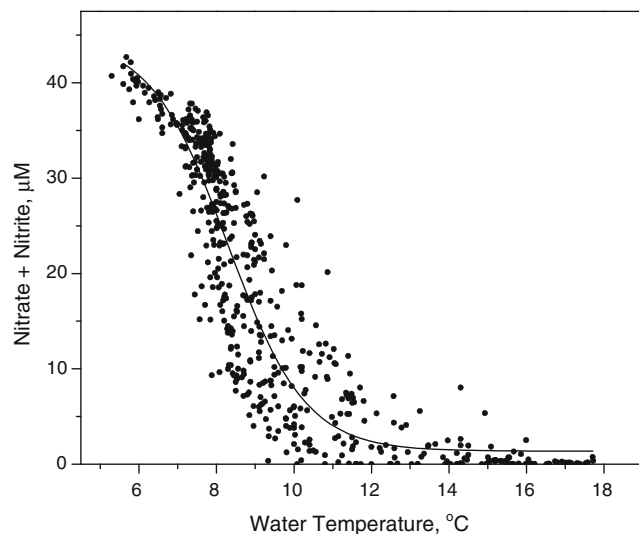


Fig. 2 Water temperature versus $\text{NO}_3^- + \text{NO}_2^-$ relationship (presented in Eq. 5) generated using dry season data for the interval of 1997 to 2004 from the inner Oregon Shelf from Wetz et al. (2005)

averaging about 2% ($n=55$). Phosphate ranged from 0.0 to 2.9 ($1.4 \pm 0.8 \mu\text{M}$, $n=463$). The flood tide concentrations of NO_3^- and PO_4^{3-} were significantly higher in 2002 than 2003 (Mann–Whitney Rank sum test, $p < 0.001$) (Table 1). There were significant correlations between nutrient concentrations collected near the surface at the OSU dock and samples in the main channel (surface and bottom samples from Station 1, in Fig. 1), with DIN concentrations about 17% lower at the OSU dock.

Potential for nutrient limitation of phytoplankton is often estimated by examining the ratio of dissolved inorganic nutrients relative to the Redfield ratio (16 mol N:1 mol P) and comparing the ambient dissolved inorganic nutrient concentrations to phytoplankton half saturation constants for nutrient uptake (e.g., Eyre 2000). Typically, if the N:P ratio of the water column $<10:1$ then phytoplankton may be limited by nitrogen and if the ratio $>20:1$, there is the potential for phosphorous limitation (Boynnton et al. 1982). In addition, if the ambient water column concentrations are less than the half saturation constants for nutrient uptake then we assume that the phytoplankton may be nutrient limited. Typical half saturation constants for DIN and DIP are 1.0–2.0 μM and 0.1–0.5 μM , respectively.

In our study, the median N:P ratio in flood tide waters (at Station OSU) was approximately 10:1 during 2002 and 2003, indicating that nitrogen would be depleted prior to the phosphorous. Of the flood tide water samples that had N:P ratio less than 10:1, approximately 50% of these samples had DIN levels greater than 10 μM and only 6.5% of the samples had DIN levels less than 2 μM . Only 3% of the flood tide samples had N:P ratio greater than 20:1 and DIP levels less than 0.5 μM . Therefore, the majority of water advected into Yaquina Bay in this summer time frame had sufficient nutrients to sustain primary productivity.

b. Chlorophyll *a*

During the summer of 2002, water column chlorophyll *a* in water entering the bay (Station OSU) from the coastal ocean ranged from 0.4 to 36 $\mu\text{g l}^{-1}$ with a mean value of 6.3 $\mu\text{g l}^{-1}$ (standard deviation = 4.6 $\mu\text{g l}^{-1}$, $n=119$). Generally, higher chlorophyll *a* levels occurred in July and August of 2002 (monthly means of 8–10 $\mu\text{g l}^{-1}$) as compared to means of 3–5 $\mu\text{g l}^{-1}$ in May, June, and September (Fig. 3). There was a negative correlation between DIN and chlorophyll *a* ($r = -0.250$, $p < 0.01$),

indicating that recently upwelled high nutrient water had low chlorophyll *a* and periods of elevated chlorophyll *a* had reduced nutrient concentrations. Data from the in situ fluorometer indicated that there was an import of oceanic chlorophyll *a* to the estuary and there was a 40% reduction between successive flood and ebb tides. Median flood tide chlorophyll *a* (from in situ fluorometer) were significantly higher than ebb tide values (Mann–Whitney Rank Sum, $p < 0.001$). During periods of import of high chlorophyll *a*, peak chlorophyll *a* levels coincided with peak salinity, demonstrating the importance of oceanic import (Fig. 4).

2. Relationship between wind forcing, water temperature, nutrients and chlorophyll *a*

The NO_3^- , PO_4^{3-} , and temperature of water (at Station OSU) entering the inlet during flood tides responded rapidly to changes in alongshore wind stress. During upwelling favorable winds, there were increases in NO_3^- and PO_4^{3-} , and concurrent decreases in water temperature (Figs. 3 and 5). During downwelling winds, there were rapid decreases in NO_3^- and PO_4^{3-} , and concurrent increases in water temperature (Figs. 3 and 5). Peak chlorophyll *a* concentrations typically occurred after periods of downwelling winds (e.g., peak on July 25, 2002).

During the dry seasons of 2002 and 2003, upwelling favorable winds occurred 70% of the time (calculated using daily average wind stress). Even though upwelling favorable winds occurred with similar frequency in 2002 and 2003, there was a difference in the character of the upwelling events. During 2002, upwelling favorable winds were sustained for long time periods (particularly during June through October), while during 2003 upwelling occurred as discrete events. During May and June of 2002, upwelling favorable winds occurred 60% of the time and these periods of upwelling were interrupted by brief periods of downwelling favorable winds. From the end of June through October of 2002, upwelling favorable winds dominated (frequency of occurrence = 74%) with a mean north–south wind stress of 0.26 dyne cm^{-2} , and the $\text{NO}_3^- + \text{NO}_2^-$ entering in flood waters remained elevated. During the interval of May to June 30 of 2002 the mean $\text{NO}_3^- + \text{NO}_2^-$ was 10.6 μM , while during July through October of 2002 the mean $\text{NO}_3^- + \text{NO}_2^-$ was 15.9 μM . During 2003, there were six discrete upwelling events (shown as shaded

Table 1 Dissolved inorganic nutrients and chlorophyll *a* entering Yaquina Bay during flood tides (grab samples from OSU) during May–October of 2002 and 2003 (Mean \pm SD)

Year	Mean $\text{NO}_3^- + \text{NO}_2^-$ (μM)	Mean NH_4^+ (μM)	Mean PO_4^{3-} (μM)	Median N:P ratio	Mean chlorophyll <i>a</i> ($\mu\text{g l}^{-1}$)
2002	12.8 ± 7.6 ($n=179$)	3.8 ± 1.8 ($n=179$)	1.7 ± 0.7 ($n=179$)	9.7 ($n=179$)	6.4 ± 5.6 ($n=120$)
2003	10.3 ± 9.4 ($n=284$)	3.4 ± 1.6 ($n=284$)	1.2 ± 0.7 ($n=284$)	11.5 ($n=284$)	4.6 ± 2.5 ($n=55$)

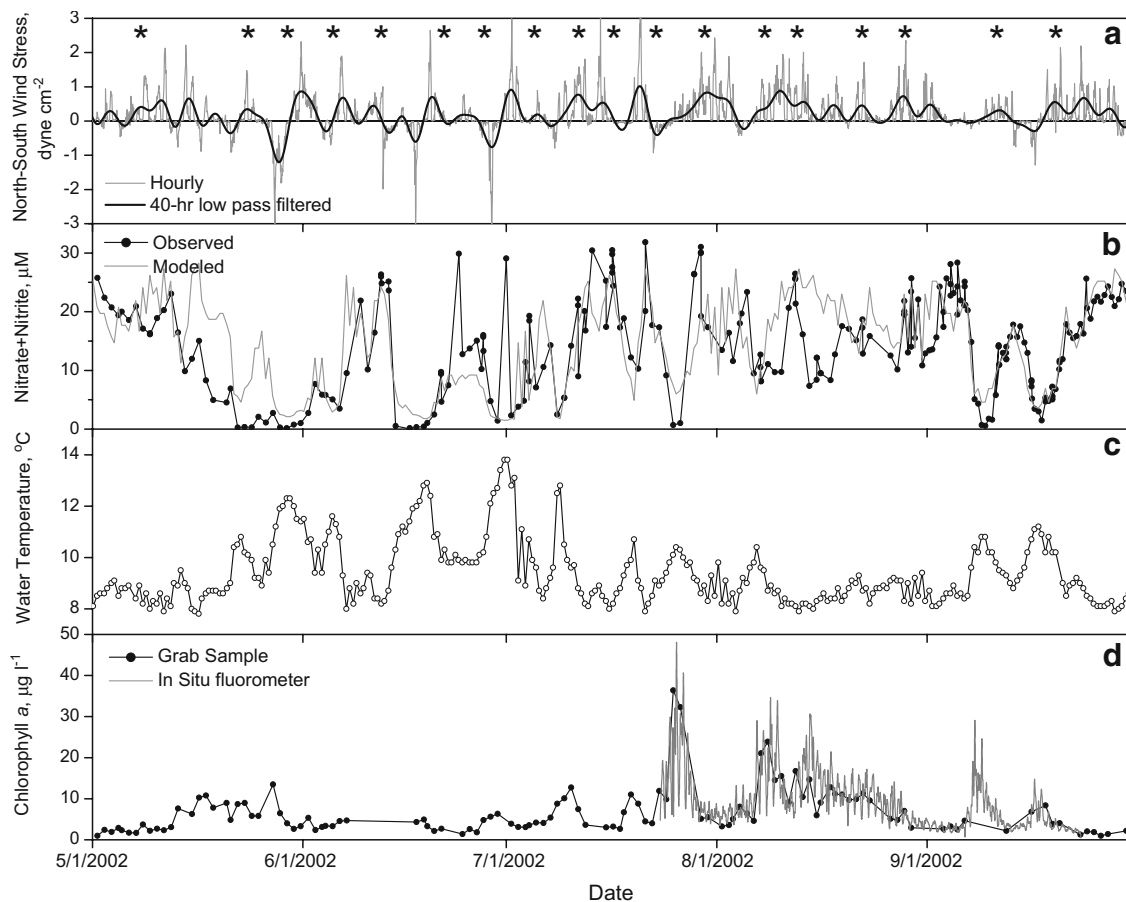


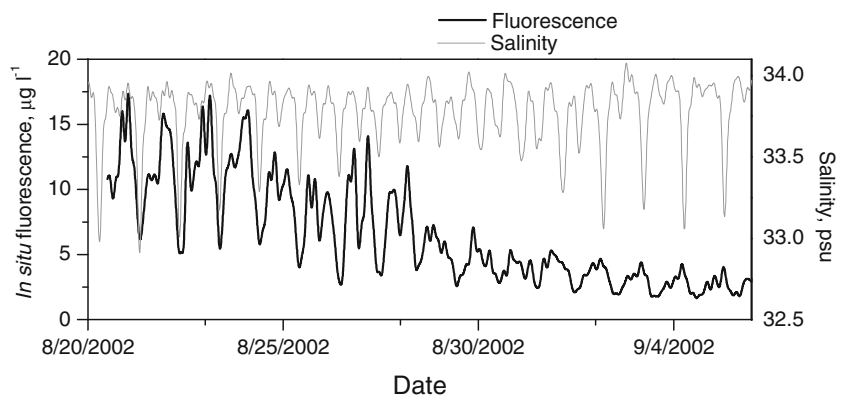
Fig. 3 Time-series of **a** north–south wind stress at NWP03 (hourly and 40-h low-pass filtered), **b** flood tide $\text{NO}_3^- + \text{NO}_2^-$ at Station OSU, **c** flood tide water temperature at South Beach, and **d** flood tide chlorophyll *a* at Station OSU (grab sample and in situ fluorometer)

during 2002. The *asterisks* (*) in **(a)** show the dates of the bay sampling. The *gray line* in **(b)** is the $\text{NO}_3^- + \text{NO}_2^-$ modeled using Eq. 5. The *filled circles* in **(d)** indicate the flood tide grab samples

regions in Fig. 5) that resulted in increases in $\text{NO}_3^- + \text{NO}_2^-$ entering the bay with each event lasting 2 to 3 weeks and peak levels during these events reaching as high as 30 μM . During 2003, the first upwelling event that caused an increase in $\text{NO}_3^- + \text{NO}_2^-$ occurred on May 27th. Between each upwelling event, there were brief periods of downwelling favorable winds or relaxation events, which lasted 1 to

2 weeks, and the $\text{NO}_3^- + \text{NO}_2^-$ levels near the end of these events were as low as 0.3 μM . During some of the upwelling events, there were brief periods (~1 d) of downwelling favorable winds that resulted in brief decreases in $\text{NO}_3^- + \text{NO}_2^-$ (such as that occurring on June 29, of 2003, Fig. 5a, b). For both years, there was a close correspondence between reversals in low-pass filtered

Fig. 4 Time-series of in situ fluorescence and salinity at Station OSU



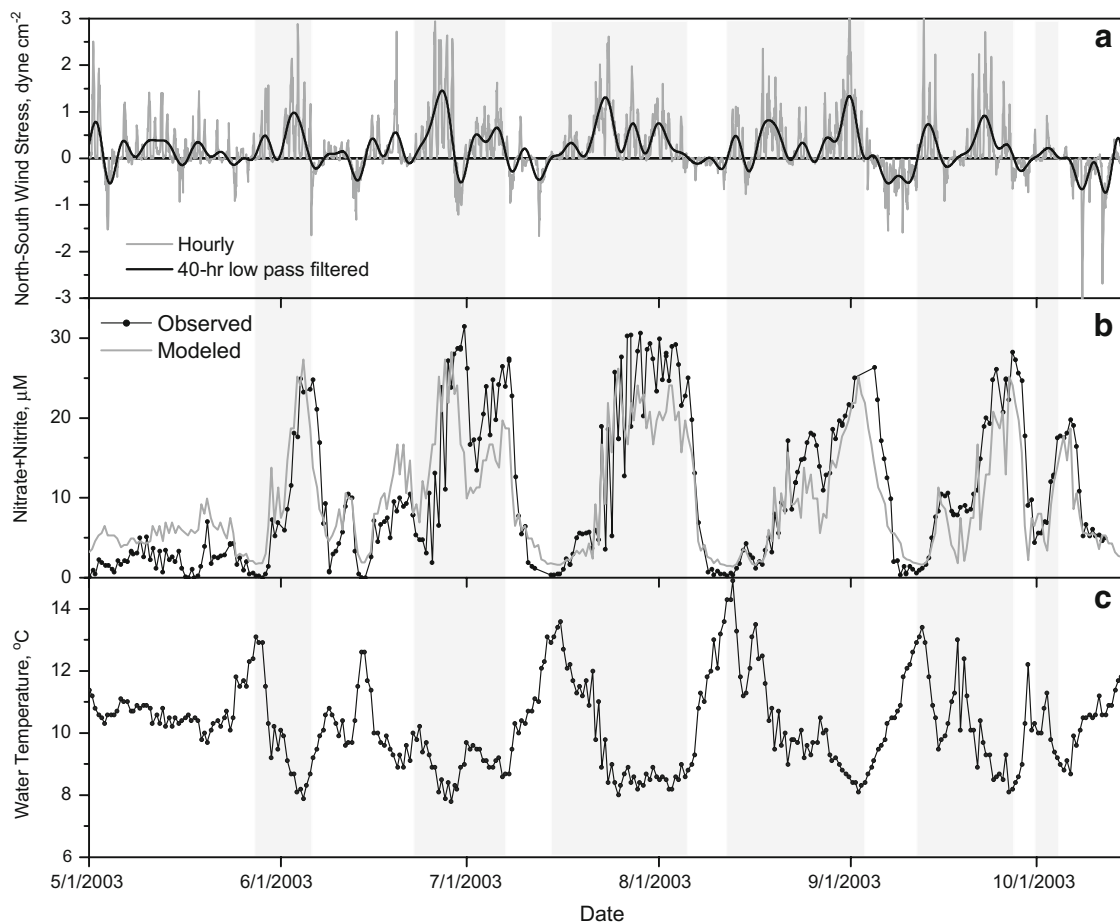


Fig. 5 Time-series of **a** north–south wind stress at NWP03 (hourly and 40-h low-pass filtered), **b** $\text{NO}_3^- + \text{NO}_2^-$ in water entering the bay, and **c** flood tide water temperature during 2003. The gray line in (**b**) is

the $\text{NO}_3^- + \text{NO}_2^-$ modeled using Eq. 5. The shaded area is upwelling events when there was a corresponding increase in $\text{NO}_3^- + \text{NO}_2^-$ input to the bay

north–south wind stress and changes in the flood-tide $\text{NO}_3^- + \text{NO}_2^-$ and PO_4^{3-} levels.

There were significant correlations between average daily north–south wind stress and flood tide water temperature, $\text{NO}_3^- + \text{NO}_2^-$, PO_4^{3-} , and chlorophyll *a* measured at Station OSU during 2002 and 2003 (Table 2). There were stronger relationships between nearshore wind stress (Station NWP03) and water column properties than using offshore wind stress (Station 46050), therefore results using nearshore (Station NWP03) wind stress are presented. The maximum cross-correlation between north–south wind stress and flood tide water temperature occurred at a lag of 1 day and the negative correlation coefficient indicates that upwelling favorable winds resulted in lower than average water temperatures. The maximum cross-correlation between $\text{NO}_3^- + \text{NO}_2^-$ and PO_4^{3-} and north–south wind stress occurred at a lag of 2 days. For 2002, the mean $\text{NO}_3^- + \text{NO}_2^-$ during upwelling and downwelling conditions was 16.1 and 8.8 μM , respectively, while during 2003 it was 12.8 and 4.6 μM , respectively. The maximum correlation

between wind stress and chlorophyll *a* occurred at lag of 6 days and the maximum correlation between water temperature and chlorophyll *a* occurred at 4 days lag; suggesting that it took approximately 5–6 days for phytoplankton to utilize the newly upwelled nitrogen and be transported across the shelf to the inlet. Although there are differences in the wind forcing between 2002 and 2003, our analysis revealed that the correlation coefficients and lags between parameters were similar in both years.

Our findings of the close coupling between alongshelf wind stress and water temperature, nutrient, and chlorophyll *a* and the lags between forcing and response are similar to previous studies. Roegner and Shanks (2001) found similar correlation and lag between wind stress and coastal and estuarine water temperature ($r=0.6$, lag 0.5–1.5 days) at Coos Bay, OR, which is located 150 km south of Yaquina Bay. Takesue and van Geen (2002) found that there was an approximately 1.5 days lag between upwelling favorable wind stress and the appearance of nearshore upwelling conditions along the Oregon coast near Coos Bay. They

Table 2 Maximum correlation between average daily north–south wind stress (computed using data from Station NWP03 with no decay coefficient), water temperature, dissolved inorganic nutrients, and chlorophyll *a* (grab samples and in situ fluorometer)

Time series compared	Max <i>r</i>	Lag (<i>d</i>)	Sample size (<i>n</i>)	Effective sample size (<i>N</i> *)
Wind stress and flood tide water temperature				
2002	−0.59****	1	294	84
2003	−0.64****	1	297	33
Wind stress and flood tide NO ₃ [−] +NO ₂ [−]				
2002	0.46****	2	179	101
2003	0.54***	2	303	31
Wind stress and flood tide PO ₄ ^{−3}				
2002	0.46****	2	179	86
2003	0.48***	2	303	40
Flood tide water temperature and flood tide NO ₃ [−] +NO ₂ [−]				
2002	−0.73****	0	173	21
2003	−0.85****	0	302	14
Flood tide water temperature and flood tide PO ₄ ^{−3}				
2002	−0.63***	0	179	17
2003	−0.72****	0	302	20
Wind stress and flood tide chlorophyll <i>a</i>				
2002	0.24**	6	109	68
Wind stress and flood tide chlorophyll <i>a</i> from in situ fluorometer				
2002	0.38**	6	120	32
Flood tide water temperature and flood tide chlorophyll <i>a</i>				
2002	−0.41**	4	113	27
Flood tide NO ₃ [−] +NO ₂ [−] and flood tide chlorophyll <i>a</i>				
2002	0.31*	7	101	26

Lag is the lag at which the maximum correlation occurs with the second variable lagging the first variable by the lag indicated

See Data Analysis section for definition of *N**

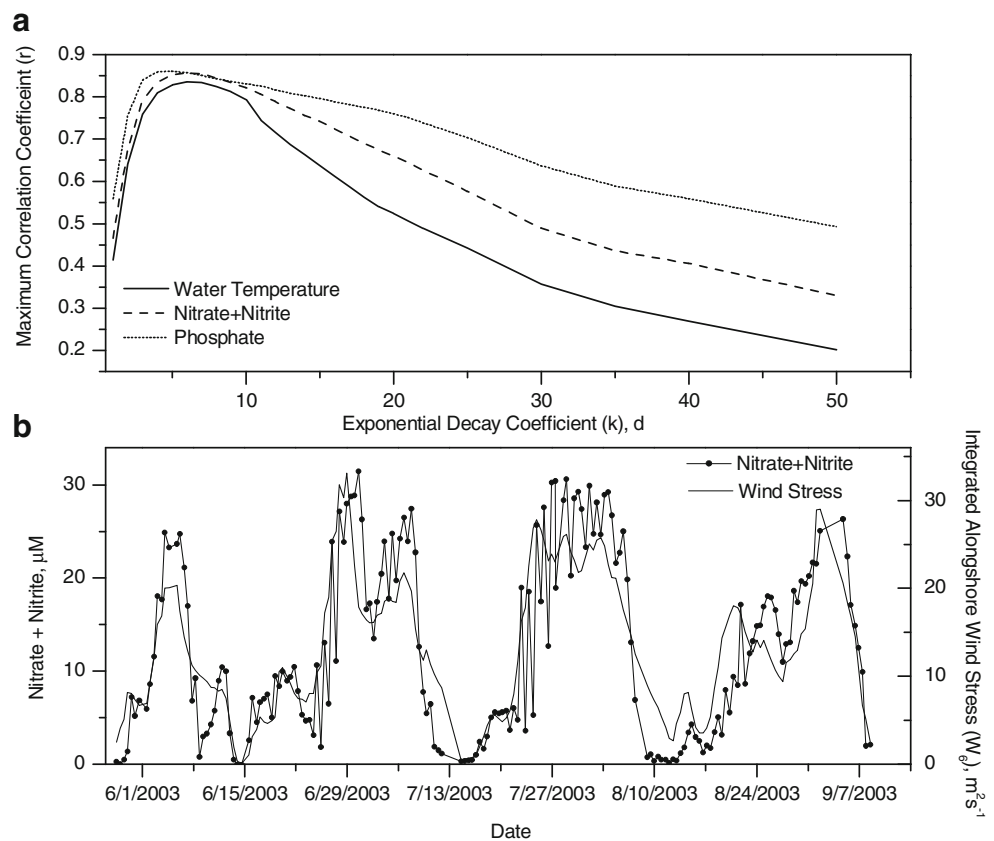
p*<0.1; *p*≤0.05; ****p*≤0.01; *****p*<0.001

found that the composition of nearshore water responds to local changes in wind forcing, which is similar to our analyses which found stronger correlations between water properties and nearshore wind forcing than offshore wind forcing. Hickey et al. (2002) found similar correlation and lag ($r=-0.6$, lag 1.25–1.5 days) between wind stress and water temperature and salinity fluctuations near the mouth of Willapa Bay, WA, USA, which is 233 km north of Yaquina Bay. Hickey and Banas (2003) examined variations in temperature, salinity and alongshore winds stress for three estuaries along the Oregon and Washington coasts, spanning 400 km. They demonstrated that there was coherence between estuarine water properties fluctuations (temperature and salinity) among these estuaries during the summer resulting from the large scale patterns in alongshelf wind forcing. However, none of these studies assessed the relationship between wind forcing and chlorophyll *a* or nutrients. Service et al. (1998) found that off of Monterey Bay, CA, USA wind stress and water temperature were maximally correlated at a lag of 2–3 days and there was a correlation between fluorescence and water temperature and wind stress at lags of 4 days and 6–7 days, respectively, which is similar to our results. Thomas and Strub (2001) performed a cross-correlation analysis between wind

forcing (longshore wind stress and wind mixing) and cross-shelf pigment variability. They found that on the shelf off of Washington and northern Oregon (including our study area) the pigment pattern metrics were poorly related to local alongshore winds. However, this is probably due to the temporal resolution of their pigment data being too coarse (10 days) to resolve the relationship between nearshore chlorophyll *a* and wind stress.

Austin and Barth (2002) developed an index of upwelling intensity based on the position of the upwelling front for the shelf off of Newport, OR, USA. They found that this index was highly correlated ($r=0.88$) with integrated alongshore windstress (W_k) with exponential decay coefficient (*k*) of 8 days, and the correlation remained strong for *k* varying from 5–12 days. In addition, they found high correlations ($r=0.7$) between nearshore (50-m isobath) temperature and salinity observations and W_8 . We found similar high correlations ($r=0.6$ – 0.8) between flood tide water temperature, NO₃[−] and PO₄^{−3} at Station OSU and W_k during 2002 and 2003. Correlation coefficients and lags were similar for water temperature, NO₃[−] and PO₄^{−3} (Fig. 6) with peak correlations ($r=0.86$) occurring at *k*=6 days. There were stronger correlations between wind stress and flood tide properties (water temperature and

Fig. 6 **a** Correlation coefficient between flood tide water temperature, $\text{NO}_3^- + \text{NO}_2^-$ and PO_4^{3-} entering Yaquina Bay and integrated alongshore wind stress (W_k) as a function of exponential decay coefficient (k) and **b** comparison of time series of flood tide $\text{NO}_3^- + \text{NO}_2^-$ and integrated alongshore wind stress (with $k=6$). The correlation coefficients for water temperature have been multiplied by -1



nutrients) calculated using nearshore (Station NWP03) wind data rather than offshore (Station 46050) for W_k . Significant correlations between W_k and nutrient concentration were obtained during both years (2002 and 2003); however, slightly stronger relationships were present during 2003. Thus, W_k may also be a useful indicator for oceanic nutrient input to estuaries in the PNW.

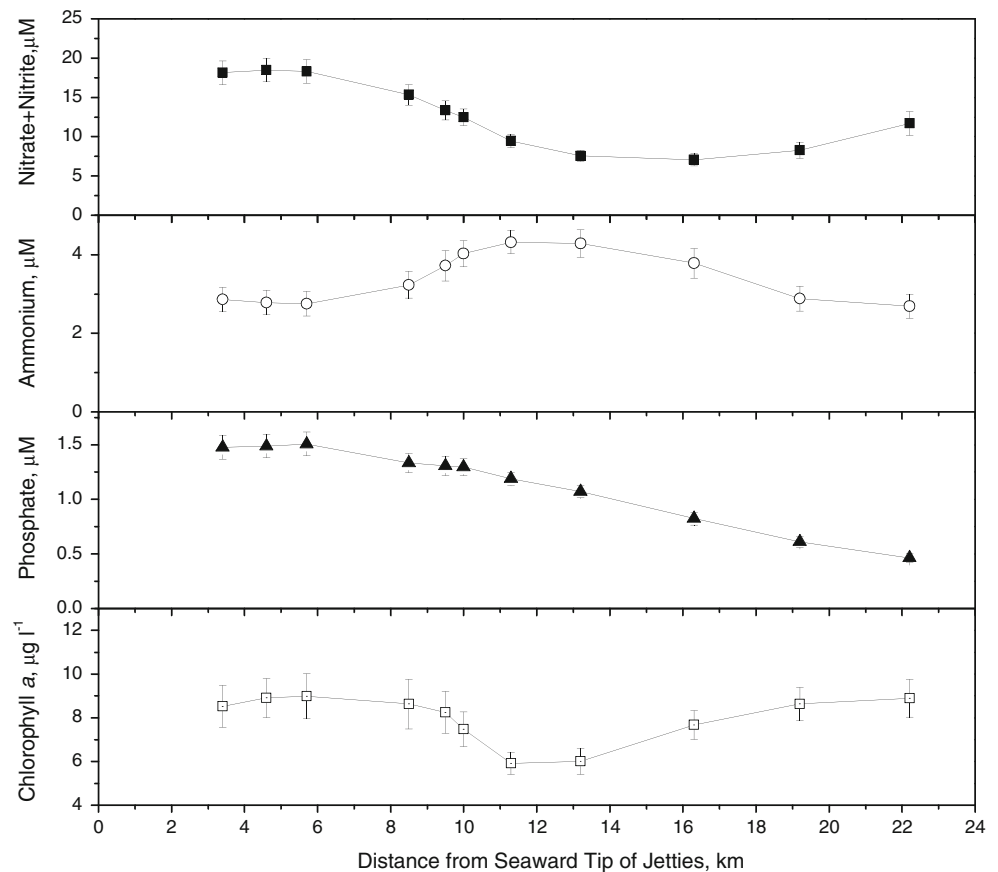
Within the Estuary Patterns During 2002

Data from the cruises were used to examine spatial patterns in nutrients and chlorophyll a within the estuary. A shift in the location of maximum $\text{NO}_3^- + \text{NO}_2^-$ concentrations in the estuary occurred during the transition from spring to summer. During 2002, from January through early June, the maximum $\text{NO}_3^- + \text{NO}_2^-$ occurred at Station 12 suggesting a riverine source for this constituent (with a mean salinity of 4.9 at Station 12 and average riverflow during this time period of $22.6 \text{ m}^3 \text{ s}^{-1}$). From January through mid April of 2002, the $\text{NO}_3^- + \text{NO}_2^-$ for the ocean boundary averaged $5 \mu\text{M}$ ($n=11$), while at Station 12 it averaged $69 \mu\text{M}$ ($n=10$) with peak concentrations of $97 \mu\text{M}$. Mixing diagrams of DIN versus salinity (not presented in this paper) revealed conservative transport of DIN during the winter. During late April of 2002, upwelling favorable wind stress resulted in the ocean boundary $\text{NO}_3^- + \text{NO}_2^-$ increasing to about $25 \mu\text{M}$.

From mid June through the end of August of 2002, the maximum $\text{NO}_3^- + \text{NO}_2^-$ and PO_4^{3-} occurred near the mouth of the estuary (Stations 1–4) suggesting an oceanic source for these nutrients. Fig. 7 shows the spatial variation in nutrients within the estuary during the dry season of 2002. There was a mid-estuary minimum in the mean dry season $\text{NO}_3^- + \text{NO}_2^-$ (mean value of $7 \mu\text{M}$, Fig. 7) suggesting that the estuary receives $\text{NO}_3^- + \text{NO}_2^-$ from both the ocean and the river. The maximum concentration of NH_4^+ typically occurred in the middle of the estuary (Stations 7–9) with a dry season mean concentration of approximately $4 \mu\text{M}$ in the middle of the estuary (Fig. 7). This mid estuary maximum in NH_4^+ is probably associated with benthic regeneration of nutrients. Benthic flux measurements in Yaquina Bay in intertidal burrowing shrimp habitat show a net DIN efflux from the benthos into the water column, primarily as NH_4^+ (DeWitt et al. 2004). Although NH_4^+ levels increased in the middle of the estuary, NO_3^- remained the dominant component of DIN (64% of DIN). The primary source of PO_4^{3-} to the system was the ocean and there was a steady decline in PO_4^{3-} with distance into the estuary (Fig. 7). There was a mid-estuary minimum in chlorophyll a (Fig. 7).

The median N:P ratio from May through August of 2002 was approximately 13:1, suggesting that nitrogen will be depleted prior to phosphorous for the majority of the

Fig. 7 Mean $\text{NO}_3^- + \text{NO}_2^-$, NH_4^+ , PO_4^{3-} and chlorophyll *a* as function of distance from the mouth of the estuary (with error bars indicating standard error and $n=39\text{--}43$) during the dry season of 2002



estuary. In late April to early May of 2002, there was the potential for phosphorous limitation in the upper portions of the estuary (Stations 11 and 12) with the N:P ratio reaching as high as 176:1. During May through August of 2002, the median DIN concentration was 15 μM , and 92% of the time the DIN was >2 μM (typical half saturation constant for phytoplankton). In only 6% of the estuarine sampling events for the dry season of 2002 was the N:P ratio <10 and DIN <2 μM , and all of these events occurred in late May to early June. In only 7% of the estuarine sampling events for the dry season of 2002 was the N:P ratio >20 and DIP <0.5 μM , suggesting the potential for phosphorous limitation in the upper portions of the estuary (Stations 11 and 12). This suggests that although the N:P ratio often falls below 16:1, the estuary was not usually limited by either nitrogen or phosphorous. This is supported by assimilation ratio data (primary production–chlorophyll *a*) of Johnson (1981) collected during the dry season near Station 10 (Fig. 1) which found that 77% of the time there were sufficient nutrients for planktonic primary production, 15% of the time there was borderline nutrient deficiency, and only 8% of the time was there evidence of nutrient depletion.

During the dry season of 2002, the oceanic signal in $\text{NO}_3^- + \text{NO}_2^-$ and PO_4^{3-} propagated approximately 13 km (measured from the seaward tip of the jetties) up the estuary

to Station 9 (see Figs. 1 and 8). During the dry season of 2002, there was a significant correlation between water column $\text{NO}_3^- + \text{NO}_2^-$ at Station 11 (19 km from mouth of estuary) and Elk City (Spearman Rank, $r=0.440$, $p=0.015$, $n=30$) suggesting that at this station the primary source of $\text{NO}_3^- + \text{NO}_2^-$ is the river. The oceanic signal attenuated more rapidly for chlorophyll *a* with a statistically significant relationship between oceanic conditions and chlorophyll *a* only evident up to Station 8 (11 km from the mouth of the estuary). Similar correlations were calculated for 2003 conditions (not presented). The import of chlorophyll *a* to the lower estuary was consistent with the findings of Karentz and McIntire (1977) that during the spring through fall seasons marine diatom genera dominated in the lower estuary (stations 3.4 and 6.7 km from the mouth of the estuary), while freshwater and brackish taxa dominated in the upper estuary (stations located 12.3 and 18.8 km from the mouth).

The more rapid decline in the oceanic signal in chlorophyll *a* compared to nutrients was probably the result of benthic grazing on oceanic phytoplankton. Oyster aquaculture is present in Yaquina Bay in the vicinity of Stations 7–9 and in the lower estuary there are tidal flats that have high densities of burrowing shrimp (DeWitt et al. 2004). Griffen et al. (2004) estimated that the daily filtration rate and density of one species of burrowing

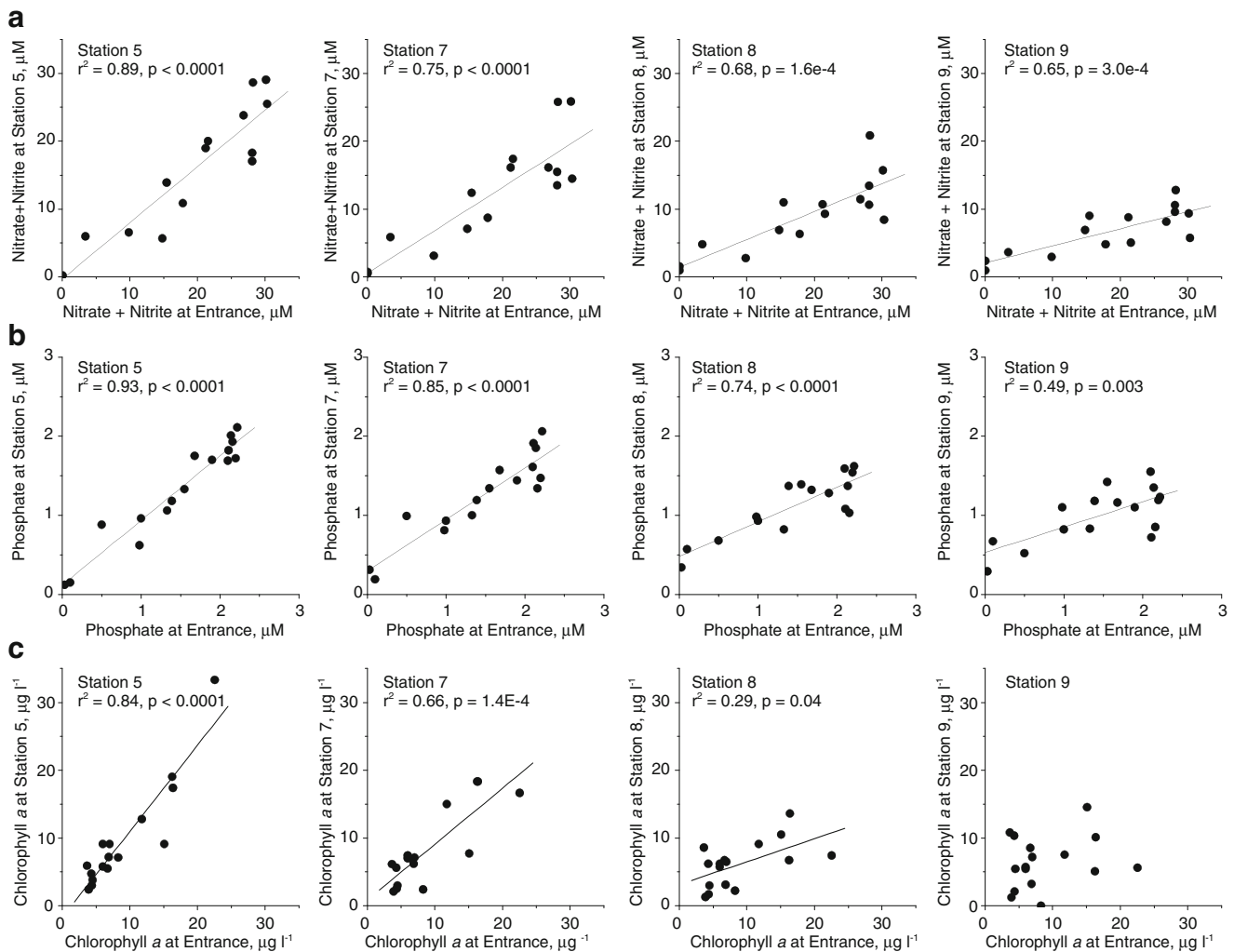


Fig. 8 Relationship between **a** $\text{NO}_3^- + \text{NO}_2^-$, **b** PO_4^{3-} , and **c** chlorophyll *a* at Stations 5, 7, 8, 9, and conditions near the entrance (Station 1) during May–September 2002

shrimp present in Yaquina Bay was sufficient to clear the entire water column of Yaquina Bay on a daily basis.

Comparison of Nitrogen Inputs

We compared the N sources to Yaquina Bay during the wet and dry seasons (Table 3). Oceanic and riverine inputs are the major N sources to the estuary with oceanic sources dominating during the dry season and riverine sources dominating during the wet season. During the dry season, benthic flux of N composes about 9% of the N inputs. Atmospheric deposition and wastewater treatment facility effluent are minor N sources.

1. Watershed inputs

There is a ninefold difference in the average daily wet season ($13.1 \text{ m}^3 \text{ s}^{-1}$) and dry season ($1.5 \text{ m}^3 \text{ s}^{-1}$) riverine discharge at Chitwood. Riverine DIN levels are related to

the discharge with wet and dry season DIN levels averaging $99 \text{ } \mu\text{M}$ ($n=44$) and $40 \text{ } \mu\text{M}$ ($n=43$), respectively (calculated using observations from Chitwood from 1979–2005). There is an order of magnitude difference in average daily riverine N input to Yaquina Bay during the wet ($2.6 \times 10^5 \text{ mol N day}^{-1}$) and dry seasons ($2.3 \times 10^4 \text{ mol N day}^{-1}$). In addition, there are considerable interannual differences in riverine N input with wet season riverine input varying from $9.4 \times 10^4 \text{ mol N day}^{-1}$ to $4.7 \times 10^5 \text{ mol N day}^{-1}$ and dry season riverine input ranging from $5.6 \times 10^3 \text{ mol N day}^{-1}$ to $6.7 \times 10^4 \text{ mol N day}^{-1}$ during the interval of 1980 to 2004. During the wet season, riverine input is the largest source of DIN to the estuary, composing approximately 74% of the input, and 92% of the annual riverine N input is delivered during the wet season. Our estimates of riverine N loading are similar to previous published values (Quinn et al. 1991; Sigleo and Frick 2007).

Compton et al. (2003) found that the presence of nitrogen fixing red alder (*Alnus rubra*) in PNW watersheds

Table 3 Comparison of nitrogen sources during wet and dry seasons for Yaquina Bay, Oregon

Source	Wet season, mol DIN day ⁻¹	Nitrogen input	
		Dry season, mol DIN day ⁻¹	Annual average, mol DIN day ⁻¹
River ^a	2.6×10^5 ($\pm 6\%$)	2.3×10^4 ($\pm 6\%$)	1.6×10^5 ($\pm 6\%$)
Ocean ^b	8.8×10^4 ($\pm 20\%$)		
2002		5.1×10^5 ($\pm 4\%$)	3.0×10^5
2003		3.8×10^5 ($\pm 5\%$)	2.3×10^5
Wastewater ^c	1.8×10^3 ($\pm 2\%$)	1.5×10^3 ($\pm 1\%$)	1.6×10^3 ($\pm 1\%$)
Benthic flux ^d	—	4.3×10^4	—
Atmospheric deposition ^e			
On estuary	2.2×10^2	1.2×10^2	1.7×10^2
On watershed	1.1×10^4	6.0×10^3	8.5×10^3

All N inputs are calculated as gross inputs to the estuary or watershed and standard errors of the mean values are provided in the parentheses

^a Average for interval of 1980–2004 with loads estimated using LOADEST program (Runkel et al. 2004) and corrected for ungauged region using Eq. 3

^b Wet season estimate based on average wet season DIN for inner shelf for the interval of 1997–2003 from Wetz et al. (2005) and modeled volume of water entering the bay; dry season estimate for 2002 and 2003 using flood tide grab samples and Eq. 4

^c Average for interval of 2002–2004 using discharge and effluent DIN data obtained from City of Toledo, Oregon

^d Published values from DeWitt et al. (2004)

^e Based on average atmospheric deposition rate for interval of 1980–2002 (Station OR02; NADP 2003) and estuary and watershed area

influences the N export from the watershed into streams. Alder is a native species in the PNW that colonizes areas disturbed by fires, logging and landslides. Compton et al. (2003) found a significant relationship between alder cover in the watershed and NO_3^- in the streams in the Salmon River watershed, which is located 45 km north of Yaquina Bay. We used two methods to estimate the contribution of red alder to riverine N loading to Yaquina Bay. Using 1996 vegetation data obtained from the Coastal Landscape Analysis and Modeling Study (<http://www.fsl.orst.edu/clams>), we estimate that about 23% of the Yaquina Watershed is vegetated with red alder (assuming that the broadleaf cover is primarily alder). Using published N fixation rates of 50–200 kg N ha⁻¹ year⁻¹ (Boring et al. 1988; and Binkley et al. 1994) and the coverage of alder in the Yaquina watershed, we estimate that >98% of the annual riverine N loading to Yaquina Bay may be related to the presence of red alder in the watershed. Compton et al. (2003) found a relationship between broadleaf and mixed cover and annual N export (N_{export} , kg N ha⁻¹ year⁻¹) in the Salmon River basin

$$N_{\text{export}} = -4.8 + 39.0P_{\text{broadleaf}} + 29.0P_{\text{mixed}} \quad (7)$$

where $P_{\text{broadleaf}}$ and P_{mixed} are the proportion of broadleaf and mixed cover in the watershed. Using Eq. 7 and the proportion of broadleaf and mixed cover in the Yaquina watershed (using the Coastal Landscape Analysis and Modeling Study dataset), we estimated that N export from the watershed is 8.6 kg N ha⁻¹ year⁻¹ and about 80% of the annual riverine N loading is related to the presence of red

alder. Thus, riverine N loading is influenced by forest species composition.

2. Wastewater treatment facility input

During the dry seasons of 2002–2004, the daily discharge of the wastewater treatment facility effluent averaged 1.6×10^3 m³ d⁻¹ and the mean effluent DIN was 972 μM . During the wet seasons of 2002–2004, the daily discharge of effluent averaged 3.6×10^3 m³ day⁻¹ and the mean concentration of DIN in the effluent was 564 μM . Annual N input from the wastewater is estimated to be 0.3% of the total N input to the bay.

3. Oceanic input

The model estimated volume of water entering Yaquina Bay during each flood tide ranges from about 1.2×10^4 m³ to 2.3×10^7 m³ due to the mixed semidiurnal tides with mean flood tide volume of 1.4×10^7 m³, which compares well to the estimated tidal prism of Shirzad et al. (1989). The volume of oceanic water entering the estuary per day averages 2.71×10^7 m³ day⁻¹.

The gross oceanic input of DIN entering the bay was estimated using Eq. 4 and flood tide samples from OSU during the dry season of 2002 and 2003. During the dry season of 2002, the amount of DIN entering the bay from the ocean during each flood tide varied from 1.3×10^4 mol N to 9.1×10^5 mol N with a mean value of 2.6×10^5 mol N, and the mean daily flood tide input of DIN was 5.1×10^5 mol N day⁻¹. During the 2003 dry season, the mean oceanic input of DIN was 3.8×10^5 mol N day⁻¹ or 25% less than 2002 dry season.

We also calculated the oceanic input of DIN during 2002 and 2003 dry seasons using the modeled water temperature versus $\text{NO}_3^- + \text{NO}_2^-$ relationship (Eq. 5). The oceanic input of DIN estimated using Eqs. 4 and 5 (calculated for each flood tide which was sampled) is 4% higher and 4% lower than estimates calculated using flood tide samples from 2002 and 2003, respectively. This suggests that the error in using Eqs. 4 and 5 to estimate DIN loading is about $\pm 5\%$. The RMSE in modeled (using Eq. 5) flood tide $\text{NO}_3^- + \text{NO}_2^-$ in 2002 and 2003 was $5.8 \mu\text{M}$ and $5.0 \mu\text{M}$, respectively. Sigleo et al. (2005) calculated the flood tide input of NO_3^- to Yaquina Bay during August of 2000 to be $13 \times 10^5 \text{ mol N day}^{-1}$, which is about triple our estimate. However, these ocean input numbers were calculated using a constant flood tide NO_3^- of $30 \mu\text{M}$.

There is substantial interannual variation in the strength and frequency of upwelling during the dry seasons. In order to estimate interannual variability in oceanic input of DIN, we examined interannual variations (during the interval of 1997 to 2003) in flood tide water temperature, flood tide concentration of $\text{NO}_3^- + \text{NO}_2^-$ (modeled using Eq. 5), and oceanic input of DIN. The estimates of ocean input for 2002 and 2003 (modeled from Eqs. 4 and 5 for all flood tides) were 4–5% less than those calculated from flood tide grab samples. During 2002, the water entering the bay was 1.3°C colder than average, flood tide $\text{NO}_3^- + \text{NO}_2^-$ concentration (modeled using Eq. 5) was 75% higher than normal, and oceanic DIN input was 45% higher than normal (Table 4). Other studies on the shelf off of Newport, Oregon found that 2002 was an anomalous year with the halocline water about 1° cooler, the nutrients (NO_3^- and PO_4^{3-}) 60% higher, and nearshore chlorophyll *a* 54% higher than in previous years (1998–2001; Wheeler et al. 2003). Thomas et al. (2003) found that during 2002 there were higher than average chlorophyll *a* concentrations over the entire shelf from British Columbia to northern California. The higher than normal nutrients resulted in increases in phytoplankton standing stock and primary

productivity and concurrent decreases in dissolved oxygen over the inner shelf (Wheeler et al. 2003; Grantham et al. 2004). These anomalous conditions during 2002 have been attributed to the advection of a Subarctic water mass (Barth 2003; Freeland et al. 2003; Kosro 2003). During 1997 and 1998, the coastal ocean and flood tide water entering the estuary was warmer than normal and there were less nutrients entering the inlet during flood tides, which corresponds to El Niño conditions in the coastal waters off Oregon (Huyer et al. 2002). Low NO_3^- and chlorophyll *a* concentrations were documented over the Oregon shelf off of Newport during this El Niño (Corwith and Wheeler 2002).

4. Atmospheric input

The atmospheric deposition rates of inorganic nitrogen along the central Oregon coast are some of the lowest in the United States with average annual deposition rate of $0.6 \text{ kg N ha}^{-1} \text{ year}^{-1}$. Atmospheric deposition of N is a minor component of nutrient inputs to Yaquina Bay with direct deposition on the Yaquina estuary only representing 0.03% of the N inputs to the estuary. In addition, atmospheric deposition on the watershed is small (8%) compared to the watershed inputs associated with N fixing red alder in the watershed.

5. Sources of uncertainty in nitrogen loading estimates

The largest source of error in the estimate of riverine loading is the limited number of nutrient samples used to generate the relationship between flow and nutrient levels; however, our estimates of riverine nutrient loading are similar to the estimate of Sigleo and Frick (2007) based on recent data with higher sample size at Chitwood and Elk City. The two largest sources of error in the estimates of oceanic loading for the dry season are that one value was used to represent the DIN for the entire flood tide and not all flood tides were sampled. Using Eqs. 4 and 5, we have estimated the interannual variability in oceanic nitrogen loading (Table 4). There is a factor of 2.4 difference between minimum and maximum dry season oceanic loading between 1997 and 2003. We have estimated the error in using Eqs. 4 and 5 to estimate oceanic loading to be about $\pm 5\%$, which is much less than the interannual variability in oceanic loading. One strength of our estimate of oceanic nutrient loading is that it estimates the loading for May–October for 2 years with a relatively high sample size. Many other estimates of ocean loading either use an average value for ocean concentration or measure the flux over a short time period (several days).

6. Comparison to nitrogen sources for other systems

The major N sources for PNW estuaries differ from estuaries in the northeastern United States. In the PNW, the

Table 4 Interannual variation in flood tide water temperature, average flood tide $\text{NO}_3^- + \text{NO}_2^-$, modeled using Eq. 5, and modeled oceanic nitrogen input (Eqs. 4 and 5) during May through September

Year	Average flood tide water temperature, $^\circ\text{C}$	Average flood tide $\text{NO}_3^- + \text{NO}_2^-$, μM	Modeled oceanic DIN input, mol N day^{-1}
1997	13.1	3.4	2.0×10^5
1998	10.9	6.1	2.7×10^5
1999	10.5	8.0	3.3×10^5
2000	10.7	7.1	3.1×10^5
2001	10.2	9.9	3.9×10^5
2002	9.4	14.6	4.8×10^5
2003	10.2	9.9	3.6×10^5
1997–2003	10.7	8.4	3.3×10^5

Table 5 Population density and forest, agricultural and urban land use in outercoast estuaries in Pacific northwest and northeastern United States

Estuary	Population density, ind. km ⁻²	Land use (% watershed area)			
		Forest	Agricultural	Urban	Other
Grays Harbor	17	91	4	2	3
Willapa Bay	5	94	3	1	2
Alsea Bay	5	98	2	0	0
Coos Bay	27	93	1	4	2
Coquille River Estuary	7	92	5	1	2
Nehalem Bay	3	98	2	0	0
Netarts Bay	36	91	0	0	9
Siletz Bay	4	97	2	1	0
Tillamook Bay	9	93	5	1	1
Umpqua Estuary	5	92	5	1	2
Yaquina Bay	14	95	2	3	0
Average for Pacific northwest	12	94	3	1	2
Delaware Bay	386	21	39	31	9
Hudson River Estuary	590	42	24	32	2
Long Island Sound	396	54	11	31	4
Narragansett Bay	425	45	6	42	7
Average for northeastern US	449	41	20	34	5

estuarine watersheds are primarily forested (mean of 94% of watershed) with agriculture and urban land use (3% and 1%, respectively) comprising a small percentage of land cover (Table 5, computed using data from the Coastal Assessment and Data Synthesis System, <http://cads.nos.noaa.gov/>). In comparison, in the northeastern US, there is a reduction in the forested land use (41%) and an increase in agricultural and urban (20% and 34%, respectively) land use (Table 5). In addition, the population density in estuarine watersheds in the PNW is low (mean=12 individuals km⁻²) compared to the northeastern United States (mean=450 individuals km⁻², Table 5). Boyer et al. (2002) found that the atmospheric deposition was the largest N source for watersheds in the northeastern U.S (averaging about 31% of nitrogen inputs), followed by net import of N in food and feed (25%), N fixation on agricultural land (24%), and fertilizer usage (15%), while N fixation on forested land only represented 5% of the inputs (Table 6). In contrast, in the Yaquina watershed, N fixation on forest lands is the dominant source and

atmospheric deposition and fertilizer usage are minor N sources (Table 6). Atmospheric N input to the Yaquina watershed is 7% of that in the northeastern US, while N fixation on forest land is five times greater (Table 6, Boyer et al., 2002). The stream N export of Nitrogen in the Yaquina watershed is comparable to catchments in the northeastern United States. The N fixation on the forest land in the Yaquina watershed is believed to be related to the presence of red alder; however, a portion of the red alder N input may be related to anthropogenic activities since there may have been changes in red alder distribution related to silviculture in the watershed.

Conclusions

The close coupling between oceanic conditions and water column constituents in Yaquina Bay during the dry season is consistent with the high degree of tidal flushing of the estuary (i.e., large tidal prism relative to volume of the

Table 6 Comparison of N inputs in Yaquina watershed to average for northeastern catchments (including catchments for Chesapeake Bay, Delaware Bay, Hudson River estuary, Long Island Sound, and Narragansett Bay)

Watershed	Atmospheric deposition kg N km ⁻² year ⁻¹	N fertilizer usage kg N km ⁻² year ⁻¹	N fixation in forest lands kg N km ⁻² year ⁻¹	Streamflow N export kg N km ⁻² year ⁻¹
Yaquina	67 ^a	116 ^b	871 ^a	1,173 ^a
Northeastern catchments ^b	959	474	167	718

^a this study

^b Boyer et al. (2002)

estuary and low river inflow). Our results are similar to a study of Boston Harbor (Kelly 1998) that demonstrated that oceanic loading can be a major source of nutrients to coastal embayments. In Yaquina Bay, approximately 60% of the estuary is located in the region where oceanic nutrient inputs dominate.

We found that there was a close coupling between local alongshelf wind stress and flood tide water temperature, NO_3^- , PO_4^{3-} , and chlorophyll *a*. The maximum cross-correlation between north–south wind stress and flood tide water temperature, NO_3^- , and PO_4^{3-} occurred at a lag of 2 days ($r=0.5$). The maximum correlation between wind stress and chlorophyll *a* occurred at a lag of 6 days. Numerous other studies have found a close coupling between alongshelf wind stress and coastal and estuarine water properties along the Washington and Oregon coasts (e.g., Service et al. 1998; Roegner and Shanks 2001; Takesue and van Geen 2002; Hickey et al. 2002; Hickey and Banas 2003), which suggests that the results from this study may be extended to other estuaries in this region.

There is considerable interannual variation in oceanic input of nutrients. In determining reference nutrient conditions for estuaries receiving nutrient inputs from coastal upwelling it is important to quantify this interannual variation in oceanic inputs. Measuring flood tide water temperature may be an inexpensive surrogate for estimating this interannual variability. In addition, the strong relationship between integrated alongshore wind stress (W_k) and flood tide nutrients may provide a means to estimate the ocean conditions during the dry season between sampling dates. Further, the seasonal shift in dominant nutrient sources to the estuaries may require establishing nutrient conditions for the wet and dry seasons.

Since all of the bay sampling was conducted during flood tides, we do not have adequate data to compute the N export from the estuary and the net N influx through the tidal inlet. The importance of oceanic input of nutrients to primary production rates within the estuary is dependent upon whether the primary producers are benthic or planktonic. We would expect that most of the oceanic input of nutrients would be exported on the subsequent tidal cycle with little utilization by phytoplankton since the transport time scales are short relative to phytoplankton uptake rates. However, in Yaquina Bay there are intertidal flats which contain benthic primary producers (seagrass, macroalgae, and microalgal mats). These benthic primary producers are inundated with oceanic nutrients twice daily during the dry season. Since these primary producers are located in the intertidal zone they are primarily exposed to flooding ocean water and consequently the gross ocean input may better represent the loading these habitats are exposed to than the net tidally averaged loading. Previous research in Boston Harbor (Kelly 1998) demonstrated that

it is important to characterize gross ocean input, not just net ocean input. We suggest this is particularly true for estuaries adjacent to coastal upwelling regions, particularly those with extensive intertidal habitats, such as estuaries in the PNW.

During the dry season, there are seasonal macroalgal blooms on the intertidal flats in the ocean dominated section of the Yaquina Bay (Kentula and DeWitt 2003). The presence of macroalgal blooms is often used as an indicator of anthropogenic eutrophication (e.g., Bricker et al. 2003). However, in Yaquina Bay macroalgal blooms in the lower estuary in the dry season may not be an indicator of cultural eutrophication due to the dominance of oceanic input of nutrients in this area. Natural abundance stable isotope data of the macroalgae in the lower estuary suggests that they are receiving nitrogen from primarily oceanic sources (unpublished data). As the next step in this research, we are using a coupled hydrodynamic and water quality model to examine how much utilization there is of oceanic versus riverine nutrients within different portions of the estuary, the importance of in situ production versus oceanic import on chlorophyll *a* patterns within the bay, and the importance of benthic primary producers and grazers on water column properties.

Acknowledgements Laura Schumacher and Chris Eide assisted with the sampling and sample analyses. Personnel from Dynamac Corporation conducted field sampling for CTD cruises and provide ISCO sampler support. We would like to acknowledge Pat Wheeler (Oregon State University) and William Peterson (National Marine Fisheries Service) for providing nutrient and water temperature data from the Oregon shelf, Lloyd Van Gordon (Oregon Water Resources Department) for providing river discharge data, and Gary Utiger of the Toledo Wastewater Treatment Facility for providing effluent data. Pat Clinton (EPA) provided GIS support and David Specht (EPA) provided YSI datasonde data. The information in this document has been funded wholly by the USEPA. It has been subjected to review by the National Health and Environmental Effects Research Laboratory's Western Ecology Division and approved for publication. Approval does not signify that the contents reflect the views of the Agency, nor does mention of trade names or commercial products constitute endorsement or recommendation for use.

References

- Austin, J.A., and J.A. Barth. 2002. Variation in the position of the upwelling front on the Oregon shelf. *Journal of Geophysical Research* 107(C11): Art. No. 3180. doi:10.1029/2001JC000858
- Barth, J.A. 2003. Anomalous southward advection during 2002 in the northern California Current: Evidence from Lagrangian surface drifters. *Geophysical Research Letters* 30(15): 8024. doi:10.1029/2003GL017511.
- Binkley, D., K. Cromack, and D. Baker Jr. 1994. Nitrogen fixation by red alder: biology, rates, and controls. In *The biology and management of red alder*, eds. D.E. Hibbs, D.S. DeBell, and R.F. Tarrant, 57–72. Corvallis, Oregon: Oregon State University Press.
- Boring, L.R., W.T. Swank, J.B. Waide, and G.S. Henderson. 1988. Sources, fates and impacts of nitrogen inputs to terrestrial

- ecosystems: Review and synthesis. *Biogeochemistry* 6: 119–159. doi:10.1007/BF00003034.
- Boyer, E.W., C.L. Goodale, N.A. Jaworski, and R.W. Howarth. 2002. Anthropogenic nitrogen sources and relationships to riverine nitrogen export in the northeastern U.S.A. *Biogeochemistry* 57/58: 137–169. doi:10.1023/A:1015709302073.
- Boynton, W.R., W.M. Kemp, and C.W. Keefe. 1982. A comparative analysis of nutrients and other factors influencing phytoplankton production. In *Estuarine Comparisons*, ed. V.S. Kennedy, 69–90. New York: Academic Press.
- Bricker, S.B., J.G. Ferreira, and T. Simas. 2003. An integrated methodology for assessment of estuarine trophic status. *Ecological Modeling* 169: 39–60. doi:10.1016/S0304-3800(03)00199-6.
- Burt, W.V., and W.B. McAlister. 1959. Recent studies in the hydrography of Oregon estuaries. *Research Briefs, Fish Commission of Oregon* 7(1): 14–27.
- Choi, B. 1975. Pollution and tidal flushing predictions for Oregon's estuaries. M.S. Thesis, Oregon State University, Corvallis, Oregon.
- Colbert, D., and J. McManus. 2003. Nutrient biogeochemistry in an upwelling-influenced estuary of the Pacific Northwest (Tillamook Bay, Oregon, USA). *Estuaries* 26(5): 1205–1219. doi:10.1007/BF02803625.
- Cole, T.M. and S.A. Wells. 2000. CE-QUAL-W2: A two-dimensional, laterally-averaged, hydrodynamic and water quality model, Version 3, Instruction Report EL-2000. U.S. Army Engineering and Research Development Center, Vicksburg, Mississippi.
- Compton, J.E., M.R. Church, S.T. Larned, and W.E. Hogsett. 2003. Nitrogen export from forested watersheds in the Oregon Coast Range: The role of N₂-fixing red alder. *Ecosystems* 6: 773–785. doi:10.1007/s10021-002-0207-4.
- Corwith, H.L., and P.A. Wheeler. 2002. El Niño related variations in nutrient and chlorophyll distributions off Oregon. *Progress in Oceanography* 54: 361–380. doi:10.1016/S0079-6611(02)00058-7.
- De Angelis, M.A., and L.I. Gordon. 1985. Upwelling and river runoff as sources of dissolved nitrous oxide to the Alsea estuary, Oregon. *Estuarine, Coastal and Shelf Science* 20: 375–386. doi:10.1016/0272-7714(85)90082-4.
- DeWitt, T.H., A.F. D'Andrea, C.A. Brown, B.D. Griffen, and P.M. Eldridge. 2004. Impact of burrowing shrimp populations on nitrogen cycling and water quality in western North American temperate estuaries. Ed. A. Tamaki. Proceedings of the Symposium on "Ecology of Large Bioturbators in Tidal Flats and Shallow Sublittoral Sediments-From Individual Behavior to Their Role as Ecosystem Engineers," p. 107–118, Nagasaki University, Nagasaki, Japan.
- Driscoll, C.T., D. Whitall, J. Aber, E. Boyer, M. Castro, C. Cronan, C. L. Goodale, P. Groffman, C. Hopkinson, K. Lambert, G. Lawrence, and S. Ollinger. 2003. Nitrogen pollution in the northeastern United States: Sources, effects, and management options. *Bioscience* 53(4): 357–374. doi:10.1641/0006-3568(2003)053[0357:NPITNU]2.0.CO;2.
- Dugdale, R.C. 1985. The effects of varying nutrient concentration on biological production in upwelling regions. California Cooperative Oceanic Fisheries Investigations Reports, Volume 26, p. 93–96, La Jolla, California.
- Eyre, B.D. 2000. Regional evaluation of nutrient transformation and phytoplankton growth in nine river-dominated sub-tropical east Australian estuaries. *Marine Ecology Progress Series* 205: 61–83. doi:10.3354/meps205061.
- Freeland, H.J., G. Gatién, A. Huyer, and R.L. Smith. 2003. Cold halocline in the northern California Current: An invasion of subarctic water. *Geophysical Research Letters* 30(3): 1141. doi:10.1029/2002GL016663.
- Graham, B.A., F. Chan, K.J. Nielsen, D.S. Fox, J.A. Barth, A. Huyer, J. Lubchenko, and B.A. Menge. 2004. Upwelling-driven nearshore hypoxia signals ecosystem and oceanographic changes in the northeast Pacific. *Nature* 429: 749–754. doi:10.1038/nature02605.
- Griffen, B.D., T.H. DeWitt, and C. Langdon. 2004. Particle removal rates by the mud shrimp *Upogebia pugettensis*, its burrow, and a commensal clam: Effects on estuarine phytoplankton abundance. *Marine Ecology Progress Series* 269: 223–236. doi:10.3354/meps269223.
- Hickey, B.M., and N.S. Banas. 2003. Oceanography of the U.S. Pacific northwest coastal ocean and estuaries with application to coastal ecology. *Estuaries* 26(4B): 1010–1031. doi:10.1007/BF02803360.
- Hickey, B.M., X. Zhang, and N. Banas. 2002. Coupling between the California Current System and a coastal plain estuary in low riverflow conditions. *Journal of Geophysical Research* 107: 3166. doi:10.1029/1999JC000160 (Art. No. C10).
- Howarth, R.W., A. Sharpley, and D. Walker. 2002. Sources of nutrient pollution to coastal waters in the United States: Implications for achieving coastal water quality goals. *Estuaries* 25(4b): 656–676. doi:10.1007/BF02804898.
- Huyer, A., R.L. Smith, and J. Fleischbein. 2002. The coastal ocean off Oregon and northern California during the 1997–8 El Niño. *Progress in Oceanography* 54: 311–341. doi:10.1016/S0079-6611(02)00056-3.
- Johnson, J.K. 1981. Population dynamics and cohort persistence of *Acartia californiensis* (Copepoda: Calanoida) in Yaquina Bay, Oregon. Ph.D. Dissertation, Oregon State University, Corvallis, Oregon.
- Karentz, D., and C. David McIntire. 1977. Distribution of diatoms in the plankton of Yaquina Estuary, Oregon. *Journal of Phycology* 13: 379–388.
- Kelly, J.R. 1998. Quantification and potential role of ocean nutrient loading to Boston Harbor, Massachusetts, USA. *Marine Ecology Progress Series* 173: 53–65. doi:10.3354/meps173053.
- Kentula, M.E., and T.H. DeWitt. 2003. Abundance of seagrass (*Zostera marina* L.) and macroalgae in relation to the salinity-temperature gradient in Yaquina Bay, Oregon, USA. *Estuaries* 26 (4B): 1130–1141. doi:10.1007/BF02803369.
- Kosro, P.M. 2003. Enhanced southward flow over the Oregon shelf in 2002: A conduit for subarctic water. *Geophysical Research Letters* 30(15): 8023. doi:10.1029/2003GL017436.
- Large, W.G., and S. Pond. 1981. Open ocean momentum flux measurements in moderate to strong winds. *Journal of Physical Oceanography* 11: 324–336. doi:10.1175/1520-0485(1981)011<0324:OOMFMI>2.0.CO;2.
- National Atmospheric Deposition Program (NRSP-3)/National Trends Network. (2003). NADP Program Office, Illinois State Water Survey, 2204 Griffith Dr., Champaign, IL 61820.
- Pyper, B.J., and R.M. Peterman. 1998. Comparison of methods to account for autocorrelation in correlation analyses of fish data. *Canadian Journal of Fisheries and Aquatic Science* 55: 2127–2140. doi:10.1139/cjfas-55-9-2127.
- Quinn, H., D.T. Lucid, J.P. Tolson, C.J. Klein, S.P. Orlando, and C. Alexander. 1991. Susceptibility and status of West coast estuaries to nutrient discharges: San Diego Bay to Puget Sound. Summary Report. NOAA/EPA. Rockville, MD.
- Roegner, G., and A. Shanks. 2001. Import of coastally-derived chlorophyll *a* to South Slough, Oregon. *Estuaries* 24: 224–256. doi:10.2307/1352948.
- Roegner, G.C., B.M. Hickey, J.A. Newton, A.L. Shanks, and D.A. Armstrong. 2002. Wind-induced plume and bloom intrusions into Willapa Bay, Washington. *Limnology and Oceanography* 47(4): 1033–1042.
- Runkel, R.L., C.G. Crawford, and T.A. Cohn. 2004. Load Estimator (LOADEST): A FORTRAN Program for Estimating Constituent Loads in Streams and Rivers. U.S. Geological Survey Techniques and Methods Book 4, Chapter A5, 69 p.

- Service, S.K., J.A. Rice, and F.P. Chavez. 1998. Relationship between physical and biological variables during the upwelling period in Monterey Bay, CA. *Deep-Sea Research Part II* 45(8–9): 1669–1685. doi:[10.1016/S0967-0645\(98\)80012-X](https://doi.org/10.1016/S0967-0645(98)80012-X).
- Shirzad, F.F., S.P. Orlando, C.J. Klein, S.E. Holliday, M.A. Warren, and M.E. Monaco. 1989. National estuarine inventory: Supplement 1, Physical and Hydrologic characteristics, The Oregon estuaries., National Oceanic and Atmospheric Administration, Rockville, Maryland.
- Sigleo, A.C., and W.E. Frick. 2007. Seasonal variations in river discharge and nutrient export to a Northeastern Pacific estuary. *Estuarine, Coastal and Shelf Science* 73: 368–378. doi:[10.1016/j.ecss.2007.01.015](https://doi.org/10.1016/j.ecss.2007.01.015).
- Sigleo, A.C., C.W. Mordy, P. Stabeno, and W.E. Frick. 2005. Nitrate variability along the Oregon coast: Estuarine-coastal exchange. *Estuarine, Coastal and Shelf Science* 64(2–3): 211–222. doi:[10.1016/j.ecss.2005.02.018](https://doi.org/10.1016/j.ecss.2005.02.018).
- Snyder, M.A., L.C. Sloan, N.S. Diffenbaugh, and J.L. Bell. 2003. Future climate change and upwelling in the California Current. *Geophysical Research Letters* 30(15): 1823. doi:[10.1029/2003GL017647](https://doi.org/10.1029/2003GL017647).
- State Water Resources Board, Salem, Oregon, 1965. Mid-coast Basin. Salem, Oregon.
- Takesue, R.K., and A. van Geen. 2002. Nearshore circulation during upwelling inferred from the distribution of dissolved cadmium off the Oregon coast. *Limnology and Oceanography* 47(1): 176–185.
- Tappin, A.D. 2002. An examination of the fluxes of nitrogen and phosphorous in temperate and tropical estuaries: Current estimates and uncertainties. *Estuarine, Coastal and Shelf Science* 55: 885–901. doi:[10.1006/ecss.2002.1034](https://doi.org/10.1006/ecss.2002.1034).
- Thomas, A., and P.T. Strub. 2001. Cross-shelf phytoplankton pigment variability in the California Current. *Continental Shelf Research* 21(11–12): 1157–1190. doi:[10.1016/S0278-4343\(01\)00006-1](https://doi.org/10.1016/S0278-4343(01)00006-1).
- Thomas, A.C., P.T. Strub, P. Brickley, and C. James. 2003. Anomalous satellite-measured chlorophyll concentrations in the northern California Current in 2001–2002. *Geophysical Research Letters* 30(15): Art. No. 8022. doi:[10.1029/2003GL017409](https://doi.org/10.1029/2003GL017409).
- Wetz, J.J., J. Hill, H. Corwith, P.A. Wheeler. 2005. Nutrient and extracted chlorophyll data from the GLOBEC Long-Term Observation Program, 1997–2004. Data Report 193, COAS Reference 2004-1, College of Oceanic and Atmospheric Sciences (COAS), Oregon State University, Corvallis, Oregon.
- Wheeler, P.A., J. Huyer, and J. Fleischbein. 2003. Cold halocline, increased nutrients and higher productivity off Oregon in 2002. *Geophysical Research Letters* 30(14): Art. No. 8021. doi:[10.1029/2003GL017395](https://doi.org/10.1029/2003GL017395).